Rewetting drained boreal peatland forests does not mitigate climate warming in the 21st century

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Abstract: Rewetting of drained peatland forests restores ecosystem functions and improves peatland ecological status. It is also considered to mitigate climate change, yet some studies challenge this view Recently, Laine et al. (2024, Restoration Ecology, 32(7), p.e14213) considered feasible restoration outcomes of boreal forestry drained peatlands and proposed that rewetting nutrient-rich peatland forests can yield immediate climate benefits. They, however, focused only on the change of soil greenhouse gas balance following rewetting. Here, we extend their analysis by including tree stand carbon sink-source dynamics, direct radiative forcing by albedo change, and broaden the system boundaries to include wood product carbon storage. We show that restoring nutrient-rich drained boreal peatland forests contributes to climate warming in short- and medium term (<200 yr), except in specific cases when tree stand carbon storage is preserved. Rewetting nutrient poor boreal peatland forests yield to persistent warming impact. Results show that ecological benefits of rewetting drained boreal forest peatlands have a climate cost, and rewetting is unlikely to mitigate climate change in the timescale commensurate with that of, e.g., EU climate goals.

Keywords: climate change mitigation, greenhouse-gas balance, forest peatland restoration, radiative forcing, rewetting, sustainability

Running head: Climate impact of restoring forest peatlands

Implications for practice:

- Restoring nutrient-rich drained boreal peatland forests to open peatland habitats contributes to climate warming in short- and medium term (<200 yr). Restoring nutrient-poor forests to open peatlands has a persistent warming impact.
- Restoring nutrient-rich drained peatlands into tree-covered mires can have cooling effect if tree stand carbon storage is preserved.
- The warming impact of restoring to open peatlands can be partly mitigated by focusing restoration activities to late rotation stands.
- Tree stand and wood product carbon dynamics dictates decadal climate impact while net carbon sequestration or loss from soil determines long-term climate impact.

1. Introduction

Restoration of boreal peatlands drained for forestry benefits multitude of ecosystem services, such as biodiversity and hydrological cycle (Elo et al., 2024; Jurasinski et al., 2024; Andersen et al. 2017, Laine et al., 2011). It is also considered to mitigate climate change (Escobar et al., 2022; Jurasinski et al., 2024), yet some studies challenge this view (Ojanen and Minkkinen, 2020). Recently, Laine et al. (2024) evaluated restoration impact through the change in the atmospheric radiative forcing (ΔRF, e.g.

Frolking et al., 2006) and proposed that restoring nutrient rich forest peatlands provides immediate climate benefits. They, however, considered only the change in soil greenhouse gas (GHG) balance following rewetting, provoking a question how robust the conclusions are if the scope is broadened to include strong carbon (C) sequestration of managed peatland forest stand, harvesting and subsequent release of C from wood products (Jurasinski et al., 2024).

In the Nordic and Baltic countries, ca. 30% of boreal peatlands have been drained for forestry during the last century (Laine et al., 2009). In Finland there are 0.6–0.8 Mha of drained peatlands, mostly nutrient poor bogs, where wood production is not economically feasible (Laiho et al. 2016; Korhonen et al. 2024). In addition, 0.76 Mha of productive drained forest peatlands are reaching the end of their 1st rotation cycle within the next decade, opening a window of opportunity to make smart decisions on their future (Korhonen et al. 2024). Managing for ecological benefits by rewetting and restoration is an option that would comply with the EU Nature Restoration law (Hering et al., 2023) but compromise wood production (Jurasinski et al., 2024). Moreover, whether restoring drained forest peatlands is synergetic or acts against reaching climate change mitigation targets of EU Climate law (Kulovesi et al., 2024) remains uncertain, yet decisions are urgent.

Draining natural peatlands for forestry has deepened water table (WT) leading to thicker aerobic layer and enhanced peat decomposition and associated carbon dioxide (CO_2) and nitrous oxide (N_2O) emissions to the atmosphere (Laine et al., 1996; Ojanen et al., 2013; Minkkinen et al., 2020). At the same time, methane (CH_4) emissions have decreased (Ojanen et al., 2013), and carbon (C) accumulation into growing tree biomass has been rapid (Minkkinen et al., 2001). While peat decomposition has accelerated in drained forest peatlands, accumulation of new C into living biomass and topsoil mor humus layer has enhanced, typically leading to net C sequestration at ecosystem level (Tong et al., 2024; Korkiakoski et al., 2023; Lohila et al., 2011; Minkkinen et al., 2001). In nutrient poor forest peatlands, also the soil can be a net C sink similarly to pristine peatlands (Ojanen & Minkkinen, 2019; Minkkinen et al. 2018). The positive climate impact of enhanced C sink after forestry drainage has been partly counteracted by decreased surface albedo (Lohila et al., 2010), but studies are consistent on the net cooling impact on global climate over the first forest rotation period after drainage (Laine et al., 1996; Minkkinen et al., 2002; Lohila et al., 2010).

After successful rewetting, hydrological functions and WT dynamics of undrained peatlands are restored, causing a cascade of biological, ecological, and biogeophysical changes that recover typical ecosystem functions of pristine peatlands. This leads eventually to stabilization of GHG balance into a new equilibrium, typically within 15–30 yr after restoration (see Escobar et al., 2022 for review). Relatively little is, however, known on peatland GHG balance dynamics after rewetting, but net soil CO_2 sink and CH_4 emissions are known to gradually increase, while N_2O emissions decrease to very low level (Escobar et al., 2022; Minkkinen et al. 2020). Overall, studies suggest that over time GHG balances return to levels comparable with undrained peatlands (Laine et al., 2019; Purre et al., 2019; Escobar et al., 2022; Minkkinen et al., 2020).

Recently, Laine et al. (2024) defined plausible restoration outcomes for drained peatland forests in Finland and showed that when nutrient-rich peatland forests are restored, their soil turns from CO_2 source to sink (Table 1), and the associated cooling is stronger than the warming caused by elevated CH_4 emissions (i.e. $\Delta RF < 0$). Laine et al. (2024) concluded that restoring nutrient-rich peatlands to forested mires yields immediate climate benefits, while climate mitigation potential of restoring nutrient poor peatlands is weak. Their results are, however, conditional to the fact that post-restoration change in only soil GHG balance was accounted for, and transition from drained to restored state assumed instantaneous. Here, we complement their analysis by including tree-stand C sink-source dynamics, approximate the direct radiative forcing caused by albedo change, and broaden the system boundaries to include the fate of wood product C storage on ΔRF (Fig. 1). We show that restoring drained boreal forest peatlands contributes to climate warming in short- and medium term (<200 yr), except in specific case on nutrient-rich peatland when tree stand C storage can be preserved.

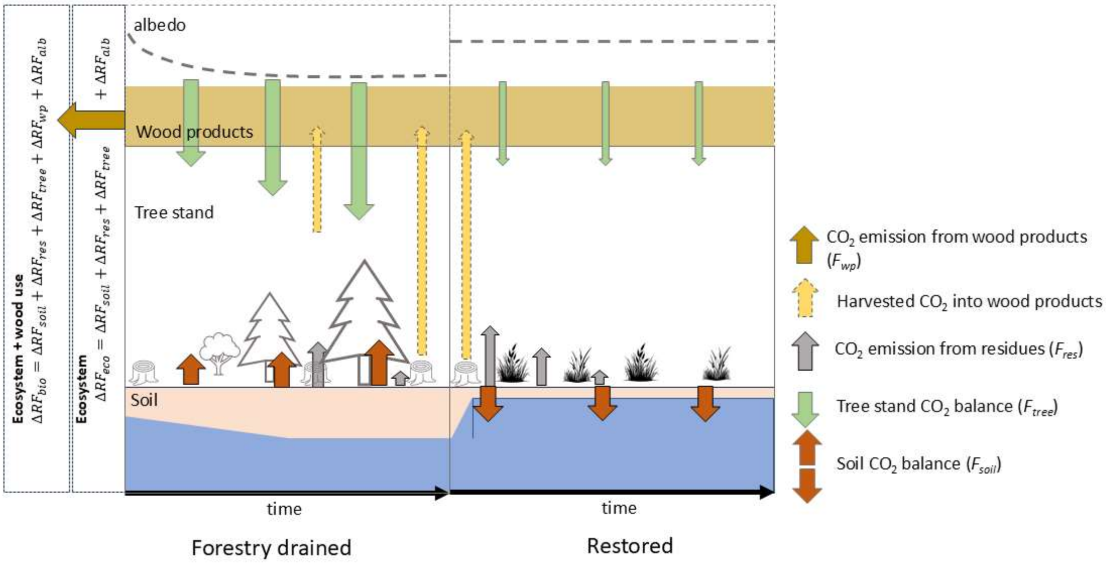

Fig. 1: Schematics on CO_2 sinks/sources (i.e. CO_2 balance) and albedo during a rotation cycle of nutrient-rich drained peatland forest, and the expected situation after restoration to open peatland.

2. Methods

2.1 Hypothetical restoration pathways

We illustrate *the effect of dynamic tree stand CO_2 sink, the fate of harvested wood products and albedo* on ΔRF using hypothetical restoration cases. *Case 1*: Restoring fertile forest peatland (FNR) in Southern Finland to open eutrophic/mesotrophic peatland. We assume restoration takes place by a clear-cut of a mature tree stand, from which the stem wood is allocated into short- and long-term wood products and harvest residues are left to decompose at the site. The restoration impacts on soil CO_2, CH_4 and N_2O fluxes ($F_{k,soil}$) and albedo change are assumed instantaneous. The wood product and residue pools are depleted during the restoration-scenario, while being periodically replenished by harvests in the reference forestry land-use scenario; *Case 2*: Restoring FNR into a spruce mire, assuming that no harvest is conducted, and tree stand C storage is preserved constant after restoration. According to Laine et al. (2024) these two restoration pathways give the strongest climate benefits, raising the question to what extent considering tree stand C sequestration, harvests and wood use would change the conclusion. In *Case 3,* we assess *how climate impact depends on timing of restoration*? We build on *Case 1*, but now initiate restoration at different times during the 58 yr forest rotation cycle. In *Case 4*, we ask *how a gradual rather than instantaneous change of* $F_{k,soil}$ from drained to restored state would affect ΔRF? For this, we assume net soil GHG fluxes change linearly over post-restoration period τ_r of up to 40 yr, covering typical equilibration time of 15 to 30 yr (Escobar et al. 2022). Finally, we compare our results to those of Laine et al. (2024). In all cases, the reference scenario to restoration is even-aged forest management, where rotation cycles and management measures are continued as in the past for the next 200 years.

2.2 Estimating the change in net GHG fluxes and atmospheric radiative forcing

Our approach utilizes, as much as possible, same assumptions and parameters as Laine et al. (2024). We use a simple book-keeping model to track the change in C stored ($S_i(t)$, kg C m^{-2}) over time (t) in drained forest / restored peatland soil, tree stand and harvest residues, and wood products made from the harvested tree biomass (Suppl. S1). The model yields annual net flux of CO_2 between the atmosphere and peatland–wood product system $F_{co2}(t)$ (g CO_2 m^{-2} a^{-1}):

$$F_{co2}(t) = F_{co2,soil}(t) - F_{tree}(t) + F_{res}(t) + F_{wp}(t), \qquad (1)$$

where negative net flux means sink of CO_2 from the atmosphere. The net soil CO_2 balance ($F_{co2,soil}$), Tree stand biomass increment (F_{tree}) and emissions from residue decomposition (F_{res}) sum up into F_{co2}, equivalent to annual net ecosystem exchange (NEE) of a peatland. Full eq. 1 includes annual emissions from wood products (F_{wp}), and thus accounts for the dynamics of the wood product C storage. We simulate biomass increment and $F_{tree}(t)$ using established forest simulator Motti (Hynynen et al., 2005), following the typical guidelines of even-aged forestry in Finland (Kellomäki, 2022). The forest dynamics is predicted for a range of site fertility types, covering the site types representing peatland forests ranging from eutrophy to oligo-ombrotrophy, and for climate conditions in the Southern and Northern Finland (see Suppl. S1.5). The mean water table depth deepens with increasing stem volume (Vol, m^3 ha^{-1}) based on Sarkkola et al. (2010) and affects net soil CO_2 balance following Ojanen and Minkkinen (2019); our formulation is a dynamic version of that used in Laine et al. (2024) to estimate soil CO_2 balance (Table 1, Fig. S4). During a stand rotation period biomass is removed in thinnings (partial harvests) and in final clear-cut and provide input to harvest residue pools that decompose at the site, and to long-term and short-term wood product (incl. bioenergy) pools. These pools emit CO_2 to the atmosphere at rates proportional to pool size and decay rate: $F_{wi}(t) = S_{wi}(t)e^{-1/\tau_i}$, where τ_i = 3 – 300 yr is the mean lifetime of pool *i*. For restored peatlands, we assume that stumps and roots decompose very slowly in anoxic conditions, $F_{tree}(t)$=0, and $F_{co2,soil}$ constant in time. For methane and nitrous oxide, we follow Laine et al. (2024) and assume only soil and forest floor processes contribute to net fluxes, which are constant in time but vary between peatland types and between drained vs. restored scenarios (Table 1). Thus, for CH_4 and N_2O, eq.1 reduces to $F_{ch4} = F_{ch4,soil}$ and $F_{n2o} = F_{n2o,soil}$.

Impact of restoration on net fluxes is computed as the difference between the net fluxes of restored (*r*) and drained (*d*) peatland, i.e. $\Delta F_{co2,d\rightarrow r}(t) = F_r(t) - F_d(t)$. We follow Laine et al. (2024) and compute how such change in net uptake/emission affects the atmospheric stocks of CO_2, CH_4 and N_2O using REFUGE 4 method (see Suppl. S1.3), which describes the dynamic response of atmospheric GHG storages to changed net surface emissions/sinks (e.g. $\Delta F_{co2,d\rightarrow r}(t)$), accounting for the aggregated effects of atmospheric chemistry and land-ocean GHG sink (Lindroos, 2023). The change in annual radiative forcing $\Delta RF_k(t)$ (W m^{-2} (earth) m^{-2} (land restored)) of gas *k*, caused by altered atmospheric concentrations, can contribute to climate warming ($\Delta RF_k > 0$) or cooling ($\Delta RF_k < 0$). The radiative forcings are additive, and the dynamic climate impact of forest peatland restoration is:

$$\begin{aligned}\Delta RF_{tot}(t) = {} & \Delta RF_{co\ ,soil}(t) + \Delta RF_{tree}(t) + \Delta RF_{res}(t) + \Delta RF_{wp}(t) \\ & + \Delta RF_{ch4}(t) + \Delta RF_{n2o}(t) \\ & + \Delta RF_{alb}(t), \qquad (2)\end{aligned}$$

where the last term ΔRF_{alb} approximates direct radiative forcing due to increased surface albedo when a forest peatland is restored to an open peatland (Suppl. S1.4). Eq. 1 – 2 enable analysis how dynamic changes in different GHG fluxes and (eco)system components contribute to $\Delta RF_{tot}(t)$.

3. Results

In *Case 1*, a fertile spruce stand at the end of its rotation (age 58 yr, Vol. ~400 m^3 ha^{-1}, mean annual increment in late rotation ~10 m^3 ha^{-1} a^{-1}; Fig. S1&S2) on mesotrophic (Mtkg) drained peatland in Southern Finland is restored to open eutrophic/mesotrophic fen (Fig. 2). Restoration contributes to climate warming (ΔRF_{tot}>0) over the first 58 yr forest rotation period and for most of the 2nd rotation cycle. Restoration starts to provide continuous climate benefits (ΔRF_{tot}<0) after the third rotation, but the average contribution remains warming for ca. 200 yr (Fig. 3 & 4). Stand productivity has significant impact on ΔRF_{tot}<0, and the more productive NRF stands are restored to open peatlands, the stronger and more persistent are associated warming impact (Fig. S3). The contribution of different (eco)system components and GHG's on ΔRF_{tot} varies over time, as their contributions are affected by stand, residue and wood product dynamics impact on net CO_2 source/sink strength (eq. 1, Fig. S1), and because of different atmospheric lifetimes of GHGs' (Suppl. S1.3). Increasing surface albedo after restoration creates persistent cooling effect, which is strongest when compared to mature forests (Fig. 2b & Fig. S4). Soon after restoration, increasing methane emissions have a major impact on ΔRF_{tot} but CH_4 effect saturates due to short atmospheric lifetime (Frolking et al., 2006). Increasing methane emissions also explain why ΔRF_{soil} is positive over the first ca. 60 – 80 yrs. After this, the role of CO_2 in radiative forcing becomes prominent and change in soil GHG balance creates cooling effect because restored peatland soil is assumed to be a constant sink of CO_2 (Table 1), while the net emissions from drained forest peatland soil continue to increase with deepening of WT towards the end of rotation (Suppl. S1.3; Fig. S4). After the first decades, variability of ΔRF_{tot} is dictated by the cyclic C sequestration / release from the ecosystem and wood products (Fig. 2a & Fig. S1). The C sequestrated into tree biomass is converted to residues and wood products after partial harvests (thinnings) and final clear-cut at the end of rotation, causing a 'saw-tooth' behavior in ΔRF_{tot} (Fig. 2a). At the end of rotation period, the forest stand CO_2 sink is temporarily removed, and rapid release of C stored in residues and wood products yields strong net CO_2 emissions (Fig. S1) into the atmosphere, causing the drop in ΔRF_{tot}.

On timescales longer than rotation period, net C sequestration into living biomass, residues or wood products is negligible, and long-term trend in ΔRF_{tot} is driven by the difference of soil C storage development between restoration (increase) and forestry (decrease in FNR, increase in FNP) scenarios (Table1, Fig. S1). This also explains the overall cooling contribution of CO_2 (Fig. 2b). Thus, the centennial dynamics of ΔRF_{tot} caused by restoring into an open peatland is driven by change in soil GHG balance, as implicitly assumed in Laine et al. (2024). The near-future climate impacts are, however, strikingly different if only soil GHG balance, or the whole system is considered.

In *Case 2*, the same forest is restored into spruce mire, now leaving tree stand intact assuming it preserves its C storage *ad infinitum* (Fig. 2c). This restoration pathway provides immediate and persistent cooling effect (ΔRF_{tot}<0), mainly because initial emissions from wood products and harvest residues from the clear-cut are avoided in restoration scenario. Also, the increase in CH_4 emissions from drained to restored state are smaller than in *Case 1* (Table 1) and resulting warming impact (ΔRF_{ch4}) remains small, and total climate impact is driven by CO_2 (Fig 2d). The effect of *Case 2* rewetting on ΔRF_{alb} is opposite to that of *Case 1*, as the albedo of mature (restored) forest stand is lower than in young (managed) stands. The difference between *Case 1* and *2* demonstrates how central the fate of pre-restoration tree stand C storage is for the climate impact.

In previous cases, restoration was done at end of rotation period in tandem with clear-cutting and regeneration (Fig. 2). In real world, rewetting a peatland area requires restoration measures are applied simultaneously at different-aged stands. In *Case 3*, we initiate *Case 1* restoration of FNR to open eutrophic/mesotrophic peatland at different times during the 58 yr rotation cycle (Fig. 3a). The results show interesting dynamics with respect to timing of restoration relative to the rotation length of managed forest: The long-term warming impact is the stronger the younger the restored stands are, as the C sink of established, growing tree stand is lost for the remaining rotation period. On the other hand, most unfavorable short-term climate impact occurs when mature forest stands are restored, as the earlier loss of biomass C storage yields to earlier large CO_2 emissions to the atmosphere compared to continued forestry scenario. When the alternative, as in this analysis, is to continue fixed-length rotation forestry, restoration to open peatland habitats will cause least climate harm if it can be done at end of rotation cycle. On peatland scale, this is, however, rarely practical.

The gradual rather than instantaneous transition of soil GHG balances from drained to restored state (*Case 4*, Fig. 3b) has only a minor effect on ΔRF_{tot}, which pales in comparison to timing of restoration (Fig. 3a) and the selected pathway (Fig. 2). This suggests that uncertainty of post-restoration GHG-balance equilibration time (Escobar et al. 2022) may not be critical for assessing climate impact dynamics, although delayed return (τ_r=40 yr) of pristine ecosystem functions (e.g. gradual increase of CH_4 emissions) seem in the studied restoration pathway lead into favorable short-term (<30 yr) and negative long-term climate impact compared to instantaneous (τ_r=0) recovery (Fig. 3b).

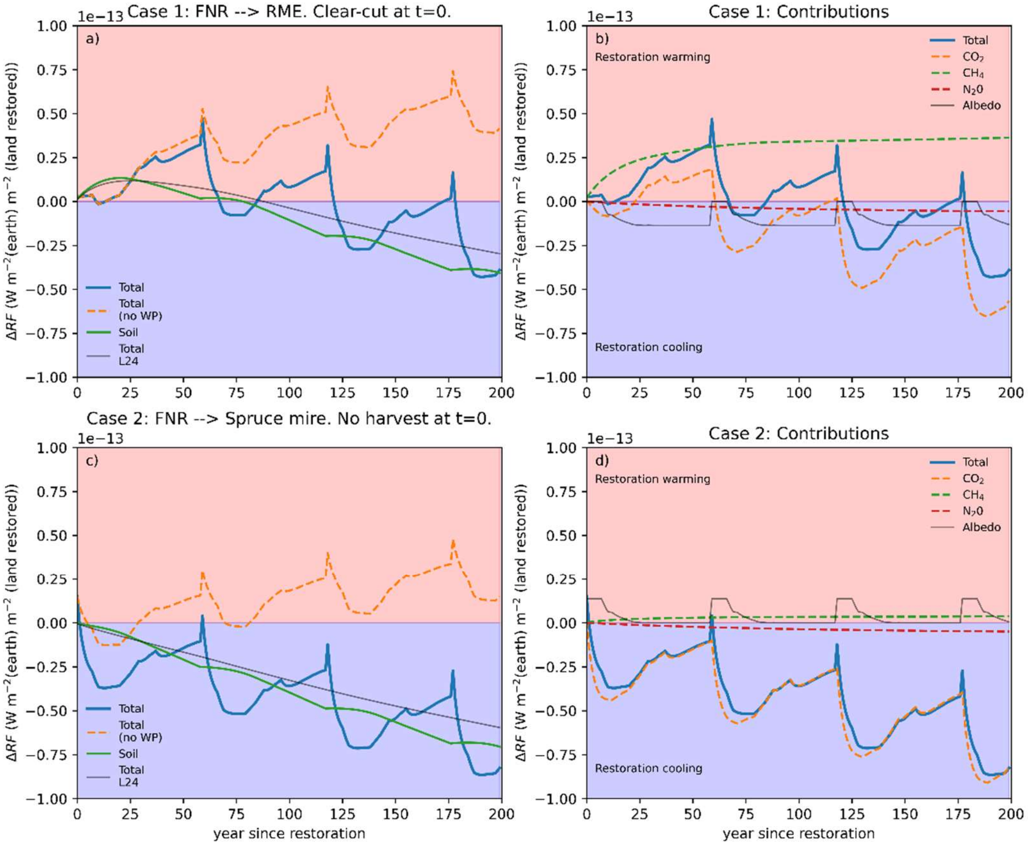

Fig. 2: Change in annual radiative forcing ΔRF_{tot} (a) and its components in: *Case 1* (b) when nutrient rich forest (FNR, mesotrophic Mtkg in Southern Finland) is restored to open eutrophic/mesotrophic peatland by clear-cutting at the end of rotation period. Development of C storages and fluxes between the system and the atmosphere are shown in Fig. S1; *Case 2* (c, d) when same forest is restored to tree-covered mire leaving the tree stand intact, assuming it preserves its C storage infinitely. In left panels (a, c) thin black line shows the estimates of Laine et al. (2024) that include only on the change in soil GHG balances. For the effect of forest productivity on *Case 1* radiative forcing, see Fig. S3.

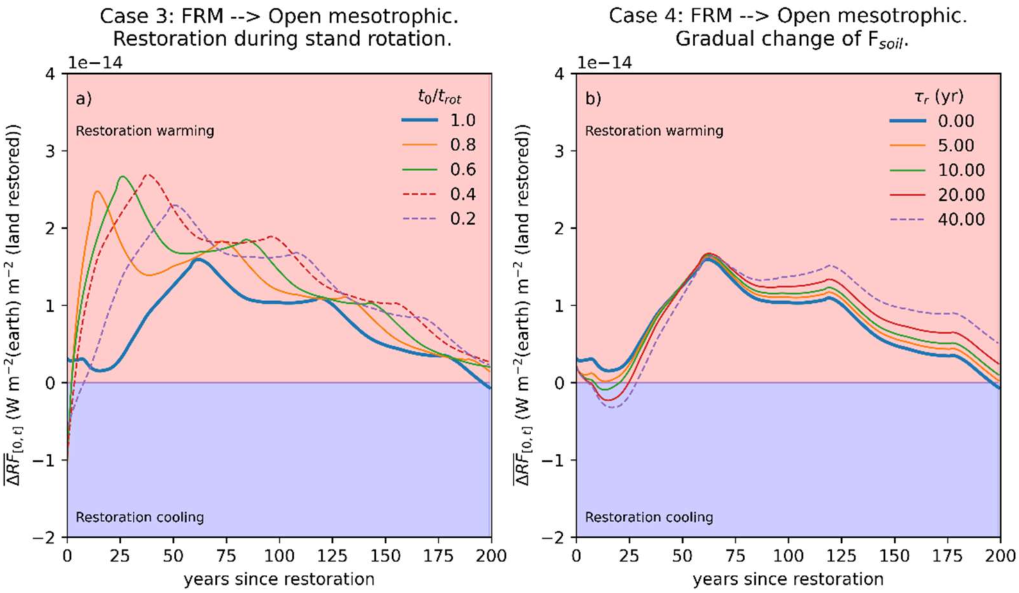

Fig. 3: Effect of timing *Case 1* restoration during forest rotation cycle (a). The continuous lines show average change in total radiative forcing ($\overline{\Delta RF}_{[0,t]}$) from time t=0 until a given point in time (in x-axis). Restoration during the forest rotation (t_{rot} < 1) leads to stronger short- and long-term warming than when restoring at rotation end. A gradual change in soil GHG balance from drained to restored state over period τ_r (b) has a minor impact on $\overline{\Delta RF}_{[0,t]}$.Thick blue line is same in both panels and equals the averaged ΔRF_{tot} from Fig. 2a.

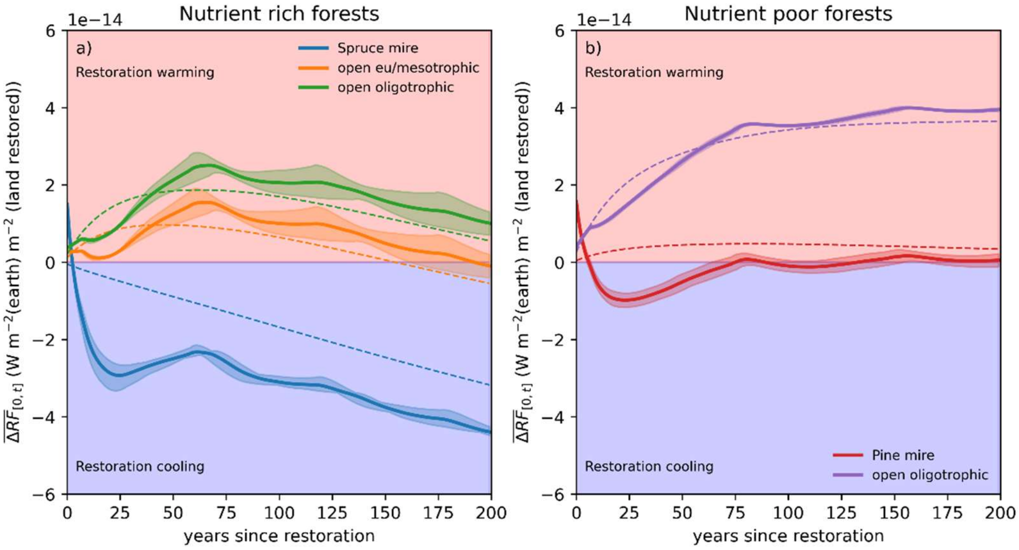

Fig. 4: Change in total radiative forcing (ΔRF_{tot}) when nutrient rich (a) and nutrient poor (b) drained peatland forests are restored to different habitats. The continuous lines show average radiative forcing ($\overline{\Delta RF}_{[0,t]}$) from t=0 until a given point in time (in x-axis). The colored range shows variability due to different forest dynamics across site-types and south-north climate gradient (see Suppl. S1.5). When restoring to open peatlands, ΔRF_{tot} includes albedo effect, while it is assumed negligible when restoring to spruce/pine mires. Vegetation C storage is assumed intact when restoring to tree-covered mires. Dashed lines show comparison to Laine et al. (2024), who included only the change of soil GHG balance.

4. Discussion

Rewetting drained boreal forest peatlands is unlikely to mitigate the climate change in the 21st century. The results unequivocally show that restoring drained forest peatlands to open peatland habitats (Fig. 2a & 4) will contribute to climate warming (ΔRF_{tot}>0) both on short term and medium term (<200 yr), while longer-term benefits may emerge when restoring nutrient rich sites. Our results align with those by Ojanen and Minkkinen (2020), who showed that restoring boreal forestry drained peatlands will have a warming effect at least for the first century after restoration, depending on forestry practices applied. Although Laine et al. (2024) only considered impacts of restoration on soil GHG balance, their results provide similar conclusion (Fig. 4). Our findings are also consistent with earlier studies showing that draining boreal peatlands for forestry has contributed to climate cooling, as C accumulation to growing tree stand has outweighed C losses from peat soil (Laine et al., 1996; Minkkinen 1999; 2002) and negative effects of decreased albedo (Lohila et al. 2010) during 1st rotation cycle after drainage.

Short to medium term (<200 yr) climate impact of restoration is dictated by the fate of the C sequestered in tree stand (Fig. 2 & 3a). If tree stand C storage can be preserved when restoring to tree-covered mires (Fig. 2b & 4), the avoided CO_2 emissions from decomposing residues and wood products gives climate benefits and it is possible to achieve the anticipated synergies between improved biodiversity etc. and climate mitigation goals (Laine et al., 2024; Dinesen et al., 2021, Bullock et al., 2011). Our results reveal that in an optimal case, successful restoration of nutrient rich forest peatlands to tree-covered mires (Fig. 4) may provide climate mitigation exceeding that offered by improved soil GHG balance only. However, our analysis also suggest that climate impacts of restoration are highly dependent on selected restoration targets (stand age, productivity, site type) and desired outcomes (post-restoration habitats), leading to varying synergies and trade-offs between different ecosystem services (Elo et al. 2024; Laine et al., 2024; Ojanen and Minkkinen, 2020). For instance, when restoration targets are open peatland habitats, the adverse short- and medium-term climate warming impact can, to some degree, lessened if restoration is applied on mature instead of young stands (Fig. 3a). Among nutrient rich forest peatlands, it is less harmful to restore low- than high-productivity stands (compare Fig. 2a,b and Fig S3).

The fate of tree stand C storage and sink, and release of CO_2 from residues and wood products determine radiative forcing dynamics at short timescales but for periods longer than stand rotation, ΔRF_{tot} trend depends on how the soil C storage develops after restoration compared to that of continued forestry use (Fig. 2a,c). This is because C storage of wood products and residues is mostly depleted during forest rotation cycles, radiative forcing caused by N_2O emissions is small overall, and that from elevated methane emissions saturates after ca. 100 yrs (Fig. 2b,d). Conclusions on rewetting climate impacts, and causal mechanisms underlying are thus highly dependent on timescale of interest. Focusing on the change in soil GHG balance (Laine et al., 2024) is viable when long-term climate impacts of restoration to open peatlands are considered but gives a biased view on short-term, and particularly when restoring to tree-covered mires (Fig. 4).

Our analysis also illustrates how the conclusion on climate impact of restoring drained peatland forests can differ depending on whether wood end-use is included (Fig. 2a,c & Fig. S3 blue line) or excluded (dashed orange line). The latter assumption is implicitly made if ΔRF_{tot} is evaluated at site level using ecosystem NEE (Fig. 1 & S1). In managed forests this would mean the harvested wood C transported from the site and turned into wood products is omitted from the analysis (or assumed to form an infinite C storage). For timescales longer than wood product life cycle this is conceptually incorrect and would unrealistically favor the forest management scenario. On the other side, rewetting a peatland is unlikely to affect regional wood demand in short-term, and restoration will create pressure to compensate for the lost wood production elsewhere (harvest leakage; Kallio and Solberg, 2018; Schwarze et al. 2002). This means the positive effects of preserving stand C storage when restoring to tree-covered mires (Fig. 2c) would be counteracted by emissions from residues and wood products caused by increased harvests elsewhere. In broader context, this means that unless restoration affects wood demand, rewetting nutrient-rich peatland forests to tree-covered mires is likely to provide only long-term climate mitigation, analogously to restoring to open peatlands (Fig. 2 & 4). Deeper exploration on roles of system boundaries is beyond the scope of this work, but results highlight the need to consider restoration gains and trade-offs as part of wider analysis and valuation of ecologically, environmentally, climatically, and economically sustainable boundaries of using forests and peatlands (e.g. Makrickas et al., 2023; Koskinen et al., 2017; Juutinen et al. 2020; Bullock et al., 2011).

Our model of forest C and GHG balance development over forest rotation cycles (Suppl. S1) assumes that forest management will continue as in the past, omitting potential benefits of changing environment, altered biogeochemistry and improved management on growth and C sequestration of drained peatlands (Hökkä et al., 2024a,b). It also neglects possible changes in future wood use. The magnitude of predicted NEE after clear-cutting of fertile forest peatland is in line with recent observations (Korkiakoski et al., 2023; Tikkasalo et al. 2024), but as we exclude ground vegetation and pioneering vegetation net primary productivity, the CO_2 sink recovery after clear-cutting is delayed compared to observations from a fertile drained peatland (Korkiakoski et al, 2023) and mineral soils (Grelle et al., 2023). Otherwise, NEE dynamics with stand age is realistic compared to those observed in managed boreal forests (Peichl et al., 2023; Goulden et al., 2011). We also omitted the possibility to adapt peatland forestry e.g. via continuous cover forestry (Nieminen et al., 2018), raising water table for better growth (Hökkä et al., 2024b) and reduced CO_2 emissions (Ojanen et al., 2010), or by lengthening rotation cycles for improved tree stand C storage. It can thus be argued that our results may unrealistically favor restoration, as future forest management on peatlands can be adjusted to improve its impact on climate.

We compared the atmospheric radiative forcing of alternative restoration outcomes to that of continued even-aged forest management. By doing so, we assume forest growth, management and wood use, as well as restored peatland GHG balance, will remain as in the past for the next 200 years. We also made a naïve assumption that tree stand C storage is preserved permanently when restoring to tree-covered mires (*Case 2*, Fig. 2c,d). These simplifications mean potential effects of increased abiotic (drought, floods, windthrows, peat fires) and biotic disturbances on peatland forests C cycle (Venäläinen et al., 2020; Lindner et al., 2014; Turetsky et al., 2004), and any changes in peatland ecosystem functions that would affect their GHG balance in future climate (Frolking et al., 2011; Wu & Roulet, 2014) are not accounted for. Our analysis focuses on the change of global atmospheric radiative forcing (ΔRF_{tot}) and does not consider biophysical impacts of rewetting on local surface energy partitioning (Helbig et al., 2020). It has, e.g., been suggested that extensive rewetting of boreal peatlands can buffers against high summer temperatures on regional scale (Helbig et al., 2020).

5. Conclusion

To comply with the EU Nature Restoration Law (Hering et al., 2023), demand to restore drained boreal forest peatlands will increase in the next decade. With limited knowledge and data on post-restoration GHG balances (see review in Escobar et al. 2022), tree growth and restoration success (Elo et al., 2024), and future peatland forest management (Hökkä et al., 2024a,b) predictions of the resulting climate impacts are well-aimed shots into the dark. Still, objective use of ecosystems ecology of managed forests and natural peatlands is our best asset to inform decision making on restoration today. Our results, supported by those of Laine et al. (2024) and Ojanen and Minkkinen (2020) show that ecological benefits of restoring drained boreal peatland forests will in most cases have a climate cost (warming impact) throughout the 21st century, acting against reaching the EU climate-neutrality 2050 target.

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Table 1: Soil GHG balances (g (gas) m^{-2} a^{-1}) used in this study. For CO_2, the rotation-cycle average of eq. S3 (Fig. S4) and range corresponding to young and mature (in parenthesis) are given. Laine et al. (2024) used constant values +265 gCO_2 m^{-2} a^{-1} (FNR) and -45 gCO_2 m^{-2} a^{-1} (FNP).

	Soil gas balance (g (gas) m^{-2} a^{-1})		
Peatland type	**CO_2**	**CH_4**	**N_2O**
Drained nutrient rich (FNR)	+384 (140…490)	+0.34	+0.23
Drained nutrient poor (FNP)	-15 (-130…+40)	+0.34	+0.08
Spruce mire	-91	+1.7	+0.10
Pine mire	-97	+4.8	+0.03
Open eu/mesotrophic	-104	+15	+0.10
Open oligotrophic	-124	+22	+0.03
Open ombotrophic	-95	+9.7	+0.03

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Supplementary material

S0: Supplementary Figures

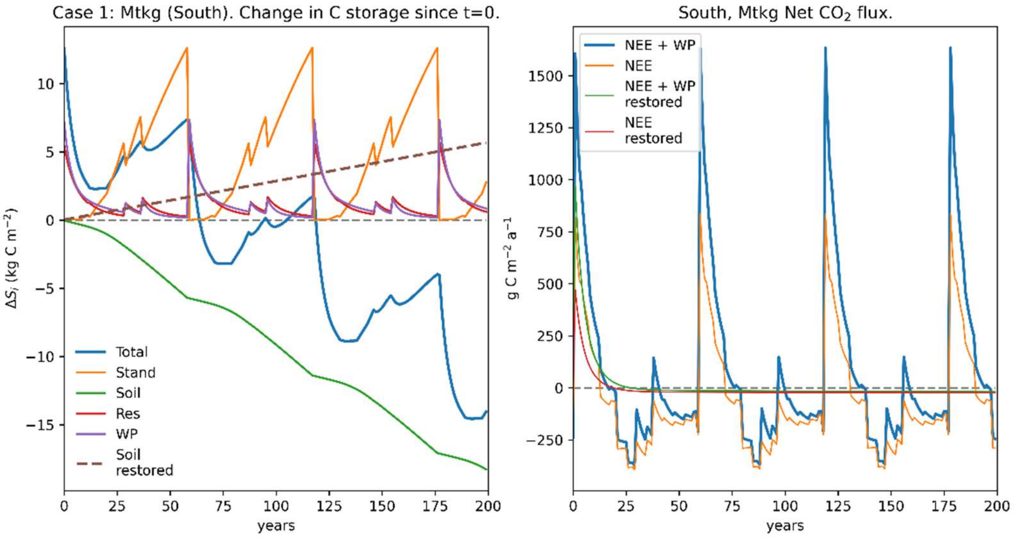

Fig. S1: Change of C storage since t=0 in continued forestry scenario in *Case 1*. The change of soil C storage in restoration scenario is shown for reference (left). The annual net CO_2 flux "felt" by the atmosphere (positive values are net emissions, right). *Stand* = living vegetation, *Res* = harvest residues left at the site, *WP* = wood products. The net ecosystem exchange (NEE) includes net stand CO_2 uptake, soil CO_2 balance and CO_2emissions from decomposing harvest residues. NEE + WP includes also CO_2 emissions from wood products.

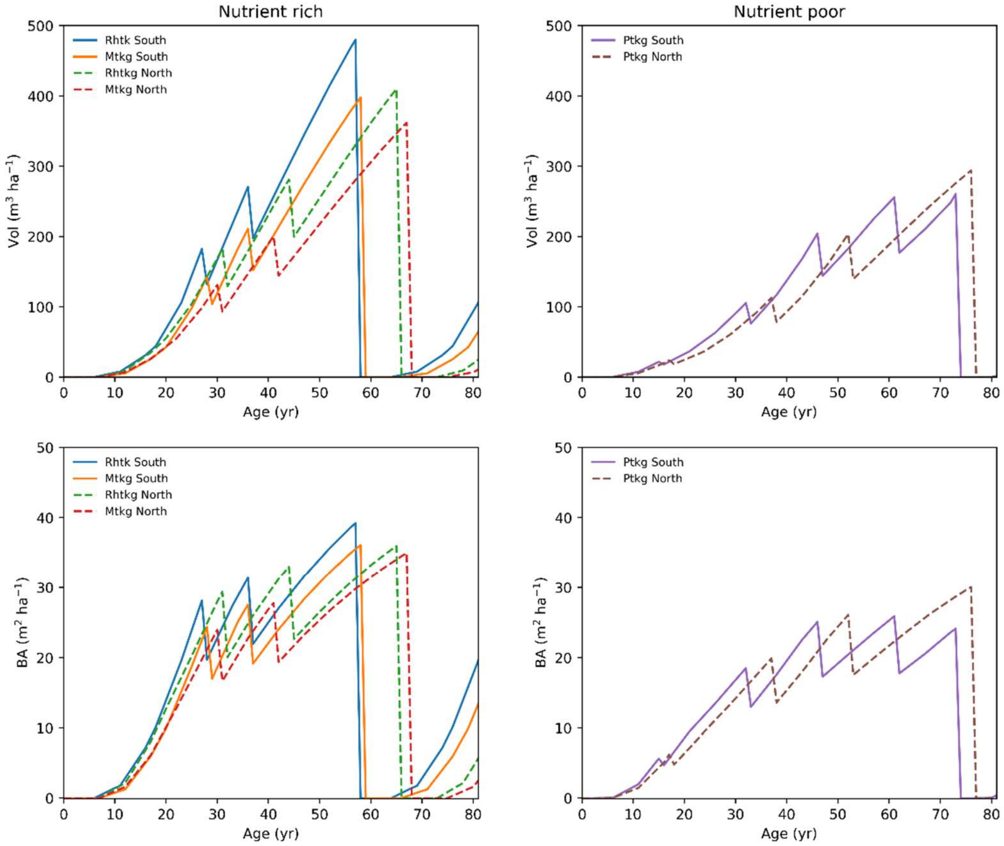

Fig. S2: Simulated development of total stem volume (Vol) and basal area (BA) over rotation cycles at nutrient rich (FNR, herb-rich Rhtkg and mesotrophic Mtkg sites) and nutrient poor (FNP, oligotrophic Ptkg site) drained peatland forests in Southern (Tampere) and Northern (Oulu) Finland. For details, see Suppl. S1.5

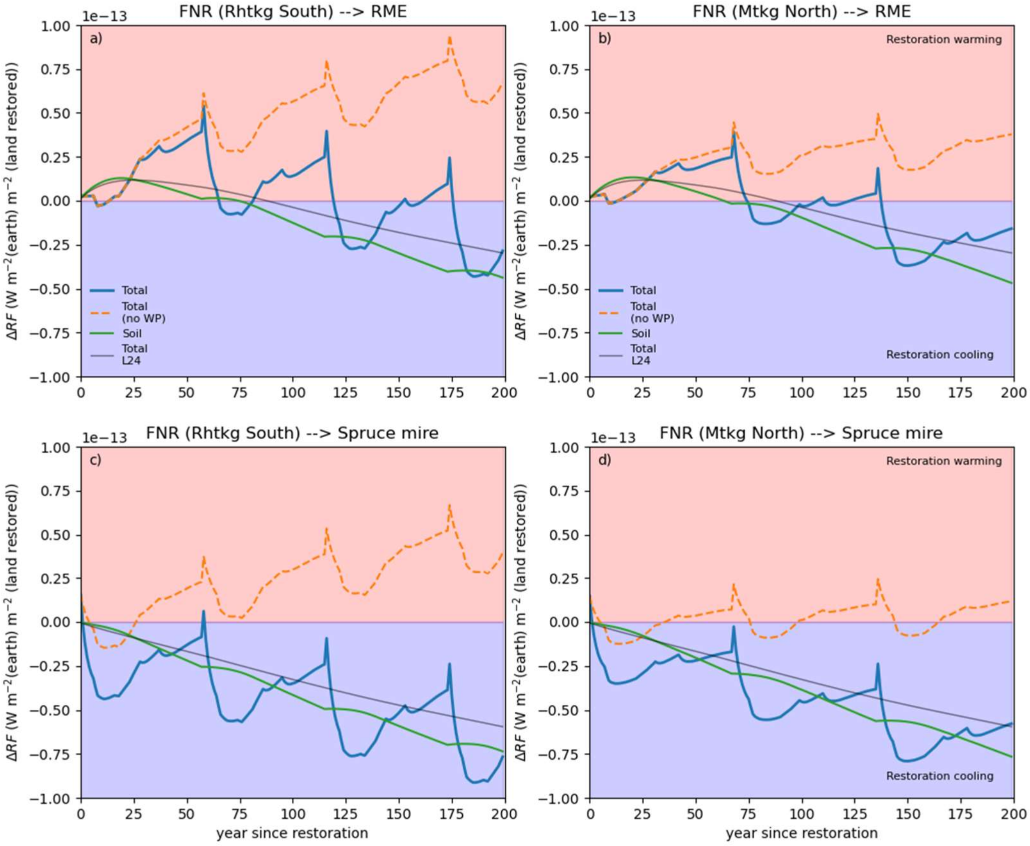

Fig. S3: As Fig. 2 but taking forest stand dynamics (Fig. S2) from a more productive herb-rich (Rhtkg, Southern Finland) and less productive mesic (Mtkg, Northern Finland) sites. Warming effect of restoring to open eutrophic/minerotrophic peatland (RME, a, b) is the stronger the higher productivity forests are restored.

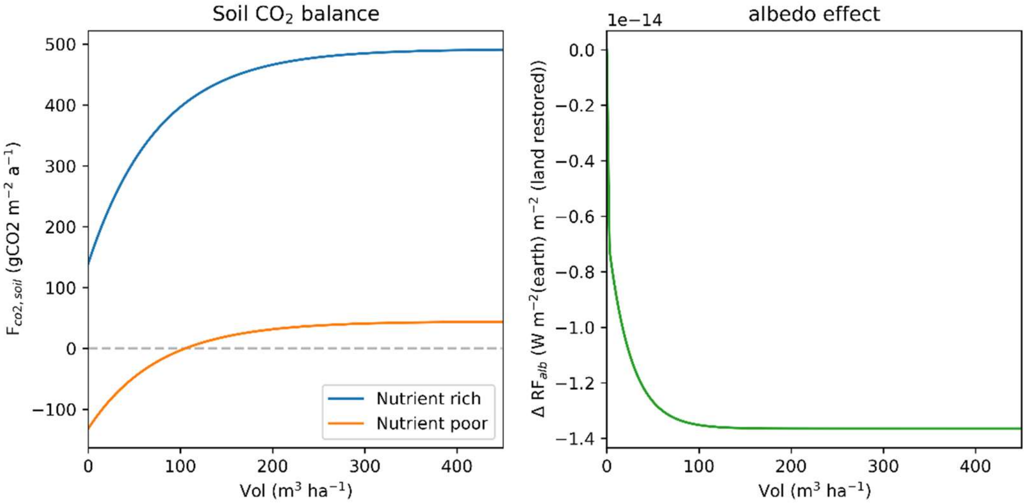

Fig. S4: Net soil CO_2 flux from drained peatland forests depends on stem volume (Vol., eq. S3; left). Approximate radiative forcing caused by albedo change (ΔRF_{alb}, eq. S14; restoration has a cooling effect) as a function of Vol. (right).

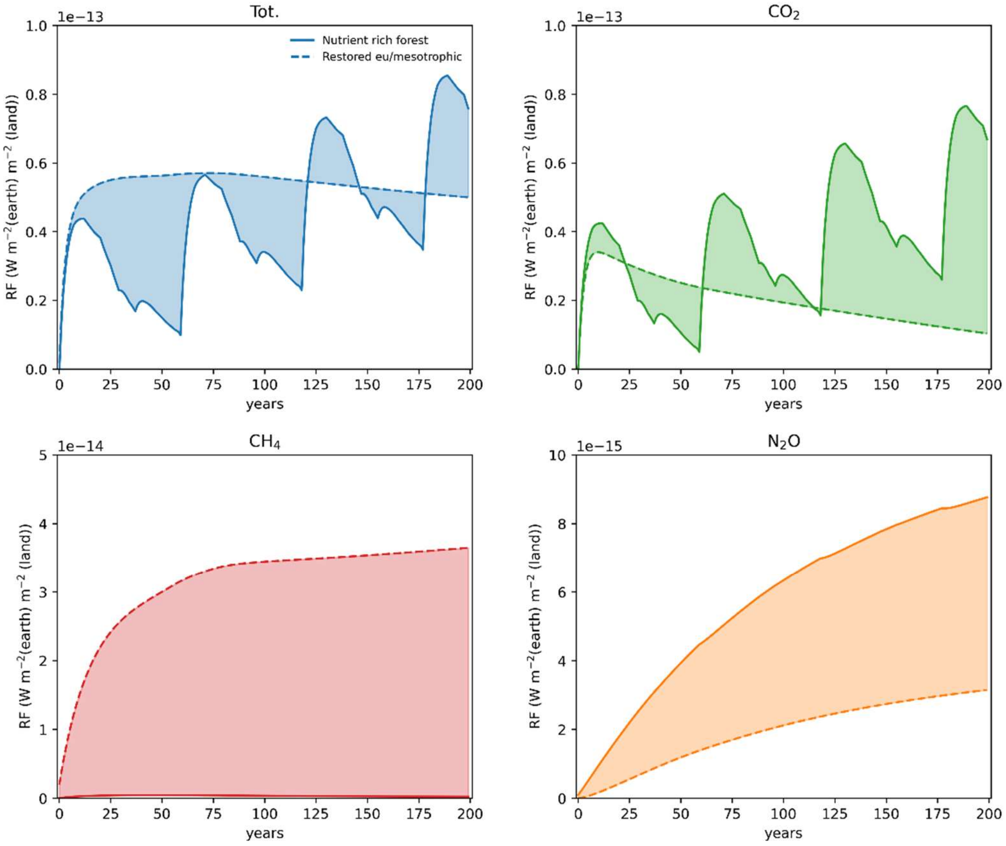

Fig. S5: Radiative forcing (RF) timeseries for nutrient rich forest (FNR, Mtkg in Southern Finland) and restored open eutrophic/minerotrophic peatland (*Case 1*, Fig. 2a,b). Total radiative forcing from greenhouse-gases (GHG's) is the sum of individual GHG's dynamic contributions (Sect. S1.3). The change in radiative forcing (ΔRF) due to restoration is the difference between restored and forestry drained peatland RF; i.e. restoration has a cooling effect when dashed line is below the continuous line. Note that y-axis scale varies between panels.

S1: Detailed methods

S1.1 Carbon balance of drained and restored forest peatland

We build a simple bookkeeping model to track the change in carbon (C) storage ΔS_i (kg C m^{-2}) in a drained forest / restored peatland and harvested wood. Total change ecosystem + wood products C storage since beginning of simulation period (t=0) is S_{bio}:

$$S_{bio}(t) = S_{soil}(t) + S_{tree}(t) + S_{res}(t) + S_{wps}(t) + S_{wpl}(t), \qquad \text{(S1)}$$

where S_{soil} refers to soil C stock, S_{tree} that of tree stand, S_{wps} and S_{wpl} in short- and long-term wood products and S_{res} in harvest residues left to the site. We evaluate change in storages (ΔS_i) and resulting net CO$_2$ fluxes ($F_{c,i}$) to/from the atmosphere at annual timestep (Δt). The ΔS_{soil} is the net soil CO$_2$ balance and varies with site fertility and water table depth (WT) according to Ojanen & Minkkinen (2019). Sarkkola et al. (2010, their Fig. 4) provide average summer WT as a function of stem volume (Vol, m^3 ha^{-1})

$$WT(Vol) = a + b \times c^{Vol}, \qquad \text{(S2)}$$

where a=-50.56 cm, b=29.36 cm and c=0.9869 are fitted parameters to their Fig.4. Ojanen and Minkkinen (2019) show $F_{c,soil}$ (g CO_2 m^{-2} a^{-1}) depends linearly on WT, while the sensitivity differs between nutrient rich (FNR) and nutrient poor (FNP) stands. Combining their eq. 3&4 with eq. S2, the soil CO_2 balance becomes a function Vol., allowing it to vary across stand rotation as (Fig. S4):

$$F_{c,soil} = d_0 - d_1 b + d_1 b \times c^{Vol}, \qquad \text{(S3)}$$

where parameters d_0 = -115 gCO_2 m^{-2} a^{-1} and d_1=12.0 gCO_2 m^{-2} a^{-1} cm^{-1} for FNR and -259 and 6.0 for FNP. Also Laine et al. (2024) estimated soil CO_2 balance based on Ojanen & Minkkinen (2019) but assumed a fixed WT and thus constant $F_{c,soil}$ over stand rotation. For restored peatlands, we assume $F_{c,soil}$ varies across peatland types as in Laine et al. (2024, Table 1) but is constant in time.

Vegetation C storage change $\Delta S_{veg} = F_{c,tree} - H$ is the balance between stand net biomass growth ($F_{c,tree}$) and periodical harvests (*H*). We estimate development of stand attributes (e.g. stem volume Vol, basal area BA and biomasses BM), and the intensity and timing of harvests, for typical nutrient rich (FNR) and nutrient poor (FNP) sites and management in Southern Finland (Sect. S1.5) using Motti forest simulator (Hynynen et al., 2005). $F_{c,tree}$ (g CO_2 m^{-2} a^{-1}) is based on the change in living biomass C storage and equals the annual net CO2 uptake from the atmosphere. For restored peatlands, we assume that annual vegetation net growth and thus change in vegetation CO_2 storage is negligible and $F_{c,tree}$=0.

The C in harvested biomass is eventually released to the atmosphere as CO_2 from decomposition of harvest residues or burning of the wood products at the end of their life cycles. The annual change in these woody pools S_{wi} is:

$$\Delta S_i(t) = \alpha_i H(t) - F(t)_{c,wi}, \qquad \text{(S4)}$$

where α_i is the fraction of harvested biomass C allocated to pool *i* (Sect. S1.5) and $F_{c,wi}$ (g CO_2 m^{-2} a^{-1}) the annual release of C as CO_2 from pool *i*. The conversion from units of C to that of CO_2 is by the ratio of molar masses. We assume wood products and residues release CO_2 to the atmosphere at a rate proportional to the pool size (1st order kinetics)

$$F_{c,wi}(t) = S_{wi}(t) e^{-1/\tau_i}, \qquad \text{(S5)}$$

where τ_i (yr) is the mean lifetime of respective pools: 3 and 20 yr for short- and long-term wood products, 3 yr for foliage and 7 yr for fine woody litter decomposing in aerobic conditions (Vavrova et al., 2009). For stumps and roots we assume τ =30 yr (Pearson et al., 2017), and order of magnitude larger when they are decomposing in anoxic conditions after rewetting.

The annual net flux of CO_2 between the atmosphere and peatland – wood product system (g CO_2 m^{-2} a^{-1}, negative values are uptake from the atmosphere) is

$$F_{c,net}(t) = F_{c,soil}(t) - F_{c,tree}(t) + F_{c,res}(t) + F_{c,wp}(t), \qquad \text{(S6)}$$

where the last two terms represent the total CO_2 release from harvest residues $F_{c,res} = \sum_{i=1}^{3} F_{c,res,i}$ and wood products $F_{c,wp} = \sum_{i=1}^{2} F_{c,wp,i}$, respectively. The change in $F_{c,net}$ and its components from drained (*d*) to restored (*r*) state

$$\Delta F_{c,i,d\to r} = F_{c,i,r} - F_{c,i,d} \qquad \text{(S7)}$$

describes the impact of restoration on atmospheric CO_2 stock $S(t)_{c,atm}$ and causes negative (cooling) or positive (warming) radiative forcing (ΔRF, W m^{-2} (earth) m^{-2} (land restored)) on global climate (Sect. S1.3). The component CO_2 fluxes (eq. 6), their changes and resulting ΔRF's are additive, which enables analyzing their separate and joint effects on climate impacts of restoration.

S1.2 CH_4 and N_2O balance of drained and restored forest peatland

Great majority of peatland methane and nitrous oxide source / sink processes take place in soils and vegetation at the soil surface (Ojanen & Minkkinen, 2019). Thus, we follow Laine et al. (2024) and assume annual net CH4 and N2O flux between the peatland and the atmosphere equal their net soil fluxes, i.e. $F_{ch4,net} = F_{ch4,soil}$, and use peatland-type specific annual values from Laine et al. (2024, Table 1).

S1.3 Radiative forcing from CO_2, CH_4 and N_2O

Changes in net uptake/emission of CO_2, CH_4 and N_2O ($\Delta F_{k,d \to r}(t)$) due to restoration affects their atmospheric stocks, and alter the radiative forcing ΔRF_k (eq. 2, W m^{-2} (earth) m^{-2} (land restored)) that can either contribute to warming ($\Delta RF_k > 0$) or cooling ($\Delta RF_k < 0$). A pulse of gas *k* emitted to the atmosphere at time t_0 creates dynamic change in $\Delta S_{a,k}(t)$, that for CH_4 and N_2O is represented as

$$\Delta S_{a,k}(t) = \Delta S_{a,k}(t_0) \times e^{-t/\tau_k}, \tag{S8}$$

where τ_k is the mean atmospheric lifetime of gas *k* (12 yr for CH_4 and 109 yr for N_2O). Emitted CO_2 pulse can undergo alternative decay pathways j in the atmosphere leading to:

$$\Delta S_{a,co2}(t) = \Delta S_{a,co2}(t_0) \times \left[\beta_0 + \textstyle\sum_{j=1}^{3} \beta_j e^{-t/\tau_{co\ ,j}}\right], \tag{S9}$$

where β_j (-) is the fractional contribution of each decay pathway, which respective $\tau_{co2,j}$ ranges from 4.3 to 394 yr (Frolking et al., 2006; Lindroos, 2023). Note that ca. 22% (β_0) of emitted CO_2 pulse accumulates in the atmosphere, explaining why reaching global CO_2 neutrality is not sufficient to reduce atmospheric CO_2 concentrations. The atmospheric lifetime functions (eq. S8 & S9) describe the dynamical response of atmospheric GHG stocks to net surface emission/sink (or their difference between restored and drained peatland, e.g. $\Delta F_{co2,d \to r}(t)$), accounting for the aggregated effects of atmospheric chemistry and land-ocean GHG sinks. After restoring a peatland at year *t*=0, the cumulated change in atmospheric stock of gas *k* at year *t* is thus the integral of each annual pulse emission/removal (Frolking et al., 2006)

$$\Delta S_{a,co}\ (t) = \int_{t=0}^{t} \Delta F_{k,d \to r}(t')\ e^{(t'-t)/\tau_k}\, dt', \tag{S10}$$

where $\Delta F_{k,d \to r}(t')$ is the restoration impact on net emissions/sinks of gas *k* on year *t'*. There are different ways to compute the change in radiative forcing $\Delta RF_k(t)$ from $\Delta S_{a,k}(t)$. For instance, Frolking et al. (2006) uses a simple and intuitive approach

$$\Delta RF_k(t) = \xi_k E_k \times \Delta S_{a,k}(t), \tag{S11}$$

where E_k(W m^{-2} (earth) kg^{-1} (gas)) is a constant radiative efficiency of gas *k*, and ξ_k (-) a multiplier for indirect effects (Ramaswamy et al., 2001). So that our results are directly comparable to those of Laine et al. (2024), we adopt the same REFUGE 4 method (Lindroos, 2023) that they used to compute mean annual radiative forcing $\Delta RF_k(t)$.

The radiative forcings from different gases and (eco)system components are additive, and the time-dependent total radiative forcing of forest peatland restoration can be written as:

$$\begin{aligned}\Delta RF_{tot}(t) = {} & \Delta RF_{co\ ,soil}(t) + \Delta RF_{tree}(t) + \Delta RF_{res}(t) + \Delta RF_{wp}(t) \\ & + \Delta RF_{ch}\ (t) + \Delta RF_{n2o}(t) \\ & + \Delta RF_{alb}(t),\end{aligned} \tag{S12}$$

where the last term ΔRF_{alb} approximates direct radiative forcing caused by increased surface albedo when a forest peatland is restored to an open peatland (Suppl. S1.4). Eq. 1 – 2 enable analysis how changes in different GHG fluxes and their components contribute to $\Delta RF_{tot}(t)$.

S1.4 Radiative forcing from surface albedo change

Restoration increases peatland reflectivity (surface albedo increases, $\Delta\alpha$) and thus the fraction of incoming global radiation ($SW_{\downarrow}$) absorbed by the land surface. The global radiative forcing caused by $\Delta\alpha$ can be approximated by (Sieber et al., 2019)

$$\Delta RF_{alb} = SW_{\downarrow} \times \Delta\alpha \times \tau_{atm} \times \frac{A}{A_{earth}}, \tag{S13}$$

where $SW_{\downarrow}$ is incoming global radiation at the surface (W m^{-2}), A = 1 m^{2} and A_{ea} = 5.1e14 m^{2} is the surface area of the Earth, and τ_{atm} (-) SW transmissivity of the atmosphere. All three first terms in eq. S13 have seasonal cycle, and $\Delta\alpha$ (forest --> peatland) depends also on forest structure, being negligible in young stands and saturate in mature stands when the canopy closes. For simplicity, we approximate annual ΔRF_{alb} using results of Lohila et al. (2010), who measured albedos of forested and open boreal peatlands and showed how ΔRF_{alb} caused by draining an open peatland for forestry varies over stand rotation of FNR and FNP forests in Finland. We digitized their Fig. 2 and 7 (FNR in Southern Finland), converted stand C storage to stem Vol using biomass expansion factors (Lehtonen et al. 2004), and fitted non-linear function to predict ΔRF_{alb} from Vol as (Fig. S4):

$$\Delta RF_{alb}(Vol) = a_0 \times (1 - be^{-cVol}), \tag{S14}$$

where *b* and *c* are fitting parameters, and parameter $a_0 = SW_{\downarrow} \times \Delta\alpha \times \tau_{atm} \times \frac{A}{A_{earth}}$ (W m^{-2} (earth) m^{-2} (land restored)) is the ΔRF_{alb} when $Vol \rightarrow \infty$ (i.e. the radiative forcing when a mature forest peatland is converted to an open peatland), and the latter term describes the shape of ΔRF_{alb}. Based on Lohila et al. (2010), a_0=-1.0e-14 W m^{-2} (earth) m^{-2} (land restored) in Southern Finland. Recently, Peräkylä et al. (2025) compared albedos and reported the difference in annually absorbed short-wave radiation (i.e. $SW_{\downarrow} \times \Delta\alpha$, their Fig. 8) at adjacent open peatland and mature forests in Southern and Northern Finland. Using average τ_{atm}=0.65 from Lohila et al. (2010), their results yield a_0=-1.36e-14 (used in this work) and a_0=-1.56e-14 W m^{-2} (earth) m^{-2} (land restored) in Southern and Northern Finland, respectively. The larger cooling effect in Northern Finland is due to later snow melt, as during winter snow-cover albedo of open peatland is much larger than that of forests, while the difference is nearly negligible in summer (Lohila et al., 2010; Peräkylä et al., 2024).

S1.5 Modeling tree stand CO$_2$ storage and allocation of harvested biomass into wood products and harvest residues

Dynamics of tree stand C storage $S_{tree}(t)$ (g C m-2) is predicted using Motti forest stand simulator (Hynynen et al., 2005; Fig. S1 & S2). Motti computes stand dynamics using linked semi-empirical models to describe the effect of species composition, tree age and size, between-tree competition, site fertility, climatic conditions, and management measures on tree growth and mortality. Stand level models are applied for predicting natural regeneration and early growth of young seedling stands until their dominant height reaches 7 m (Hynynen et al. 2014). Size distribution models (Siiplehto 2006; Siipilehto & Mehtätalo 2013) are then applied to produce tree-lists, after which individual-tree models are used for predicting growth (Hynynen et al. 2002). Tree level models have been calibrated against long-term growth experiments and National Forest Inventory (NFI) measurements, and model predictions compare well with typical stand dynamics. In Finland, typical rotation period on drained peatlands is 50–100 yr depending on site type and climate, and management goals, and contains 1–3 thinning's and a final felling.

We predicted development of stand attributes and harvest dynamics (e.g. Fig. S1) for FNR and FNP forests as follows: Starting from stand regeneration by planting or seeding, we simulated stand development (e.g. stem volume (Vol), basal area (BA), dry biomass and its components) until final felling. Regeneration and management measures such as thinning's and ditch network maintenance were done following current practices in peatland forestry in Finland (Vanhatalo et al. 2019). For FNR, we consider spruce on eutrophic herb-rich Rhtkg and mesotrophic/Vaccinium myrtillus type Mtkg site types, while FNP forests are represented by pine on oligotrophic/Vaccinium vitis-idaea type Ptkg type. To cover climatic variability, we simulate growth rates and $S_{tree}(t)$ both for Southern Finland (Tampere, temperature sum 1226 dd°C) and Northern Finland (Oulu, 1086 dd°C). In Fig. 2–3, we use mesotrophic Mtkg stand in Southern Finland as case-example, while the effect of growth and $S_{tree}(t)$ variability due to site-type and climate is accounted for in Fig. 4 (and in Fig. S3).

We compute S_{tree} from simulated total above- and below ground dry biomass assuming half of dry biomass is C. The $F_{c,tree}$ does not include ground vegetation and likely underestimates vegetation net primary production (NPP) during early stages of stand development (e.g. Peichl et al., 2023). When stand is harvested, we assume harvested C in biomass is transferred into long- or short-term wood products and harvest residues left to the site (Fig. S1). Harvested stem biomass is allocated to long-term wood products (timber, plywood) based on sawlog fraction (f_{log}, available from Motti simulations) of total stem C multiplied by saw yield (α=0.4). The remaining part of stem biomass moves into short-term wood products pool (e.g. fibre, paper, cardboard, bioenergy).

Fig. S1 illustrates C storage dynamics and net fluxes to the atmosphere when wood products are included/excluded from the calculations. Example is FNR stand (mesotrophic Mtkg) in Southern Finland, with total wood production of 525 m^3 over the 58 yr rotation. Fig. S2 shows development of key stand attributes in the same simulation.

S1.6 Restoration pathways

Laine et al. (2024) divided drained forest peatlands into two classes, which is followed here: The nutrient rich forests (FNR) include eutrophic herb-rich (Rhtkg) and mesotrophic/Vaccinium myrtillus type (Mtkg) site types. Before drainage for forestry, they have been forested or sparsely forested spruce or pine mires with admixture of pubescent birch. Currently they are dominated by Norway spruce and belong to the most productive forests in Finland; the mean annual increment in late rotation stand varies from 8 to 11 in Southern Finland and from 6 to 8 m^3 ha^{-1} a^{-1} in Northern Finland (National Forest Inventory; Korhonen et al., 2024). These net growth rates correspond to net CO_2 uptake ($F_{c,tree}$) of 1050–1450 gCO_2 m^{-2} a^{-1}. Among the nutrient poor (FNP) drained forest peatlands, oligotrophic/Vaccinium vitis-idaea type (Ptkg) is particularly important for wood production. The Scots pine is dominant species with corresponding late-rotation mean annual increment from 4 to 5 m^3 ha^{-1} a^{-1} ($F_{c,tree}$ 500–750 gCO_2 m^{-2} a^{-1}). FNP has its origin in nutrient-poor bogs or poor oligotrophic pine fens.

We explore following restoration pathways: FNR stands can become tree-covered spruce mires, open eutrophic/mesotrophic or oligotrophic mires. FNP stands can be restored to pine mires or open oligotrophic mires.

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